

An Analysis of Stream Habitat Conditions in Reference and Managed Watersheds on Some Federal Lands within the Columbia River Basin

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Abstract.—The loss of both habitat quality and quantity for anadromous fish in the Columbia River basin has been identified as a major factor in the decline of many species and has been linked to a variety of land management activities. In this study, we compared stream reaches in watersheds representing both managed and reference conditions to determine whether we could detect differences in physical habitat variables. We divided stream habitat measures into three components: stream banks, instream habitat (pools and pool depth), and stream substrate. We randomly sampled perennial streams within 261 sixth hydrologic unit code (HUC) stream reaches on federal lands in Idaho, Montana, Oregon, and Washington. The sample population represented stream reaches in 62 reference watersheds and 199 managed watersheds. An unbalanced, incomplete-block-design analysis of covariance (ANCOVA) was performed on each of the habitat variables using geology type as the block effect and bank-full width, stream gradient, and average precipitation as the covariates. Watersheds containing reference stream reaches had a slightly higher percentage of federal lands, were smaller, tended to occur at higher mean elevations, and received more annual precipitation than did the managed watersheds. We observed differences in most measures of stream habitat between reference and managed watersheds, generally in the direction we expected. Stream banks were more stable and more undercut in reference stream reaches. Pools in reference stream reaches were deeper than pools in managed stream reaches and had less fine sediment in pool tails. Analysis of covariance was an effective way to compare data across a large, relatively heterogeneous landscape where sample site stratification may be impractical or sample sizes are limited. We believe that the comparison of reference conditions to conditions across managed landscapes provides a credible way to report on the condition of these systems in lieu of trend information at individual sites.

The decline of native fish species in western North America has prompted renewed interest in monitoring the relationships between land management activities and aquatic and riparian ecosystems. The loss of both habitat quality and quantity for anadromous fish in the Columbia River basin has been identified as a major factor in the

decline of many species (U.S. Forest Service and Bureau of Land Management 1995). Improvement in stream habitat has been recommended as one of the primary steps in the recovery of fish populations within the basin (National Marine Fisheries Service 1995; U.S. Forest Service and Bureau of Land Management 1995; Kareiva et al. 2000)

The decline of stream habitat quality and quantity in the basin has been linked to a variety of land management activities including livestock grazing, road construction, agriculture, urbaniza-

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tion, timber harvest, and mining. Changes in stream habitat are generally thought to be a consequence of improper management or implementation of these activities. The consequences of these activities are often reported as changes in stream habitat, water quality, hydrology, riparian vegetation, and aquatic biota.

The suite of physical changes that result from land use impacts are manifested in three major components of stream habitat: the stream banks, the stream channel, and the stream substrate. Studies examining the influence of livestock grazing have used variables that measure changes in stream banks, including bank angle (Platts 1991; Knapp and Matthews 1996), the percent and depth of undercut banks (Myers and Swanson 1995; Knapp and Matthews 1996), and bank stability (Kauffman et al. 1983; Platts and Nelson 1985; Myers and Swanson 1992). Variables used to evaluate changes in the stream channel include the width of the stream (Kauffman et al. 1983; Platts et al. 1983; Matthews 1996); the change in the distribution, type, and morphology of channel units (Marcuson 1977; Hubert et al. 1985; Myers and Swanson 1996); and changes in channel form (Marcuson 1977). Changes in substrate include increases in fine sediment (Hubert et al. 1985; Myers and Swanson 1996), reduced spawning habitat (Duff 1983), and changes in riffle particle sizes (Kappeser 2002).

Studies evaluating changes in stream habitat from logging, road construction, mining, and other activities have used variables similar to those used in grazing studies, but may be more focused on the consequences of a particular activity. For example, the consequences of poor timber harvest practices or improper road design or construction may be more related to changes in large woody debris and sediment supply. Variables that have been used to assess these changes have included the amount and depth of large pool habitat (Woodsmith and Buffington 1996; McIntosh et al. 2000), the volume of fine sediment in pools (Lisle and Hilton 1992), riffle armoring (Kappeser 2002), and the alteration of large woody debris input (Bisson et al. 1987; Woodsmith and Buffington 1996).

Studies of land use effects have been conducted at a variety of scales. Grazing effects studies typically involve a comparison of riparian and stream habitat changes that occur when cattle are excluded from certain sections of streams (Duff 1977; Platts 1991; Magilligan and McDowell 1997). Long-term differences in riparian and stream habitat within an exclosure can serve as a reference

point for the direction and rate of recovery of these systems once livestock are removed. Disadvantages of these studies are that they are often of limited value because the exclosures are too small, poorly located, or not replicated (Rinne 1999; Sarr 2002).

Long-term, whole-watershed studies can be used to track the consequences of a management manipulation through a stream network over a longer time. These studies provide insights into the extent of the perturbation(s) and the duration of the effect on the stream. For example, long-term (>20 years) studies in Carnation Creek, British Columbia, indicated that logging and road building have increased landslides and debris torrents that have altered stream channels and habitat, ultimately reducing the quality of spawning and rearing habitat for salmonids (Hartman et al. 1996). A potential drawback of these types of studies is that results may not be applicable to broader scales. Although some inferences about changes in process and function can be made, differences in climate, elevation, precipitation regime, and vegetation type may limit the ability to extrapolate results from single sites to other areas.

Studies that are conducted across multiple stream reaches or watersheds may provide insights at a broader scale (Ralph et al. 1994; Woodsmith and Buffington 1996). Woodsmith and Buffington (1996) compared stream habitat in watersheds with extensive timber harvest and watersheds with no harvest in 23 forest stream reaches in southeast Alaska. They found that they could discriminate their "reference" watersheds from the managed watersheds by differences in three measures of stream habitat: total numbers of pools per reach, ratio of mean residual pool depth to mean bank-full depth, and the ratio of critical sheer stress of the median surface grain size to bank-full sheer stress. Variability and range in pool frequencies were unchanged in streams representing "natural" streams, but decreased during the 1930s to the 1990s in the Columbia River basin (McIntosh et al. 2000). Inferences from these types of studies are often extrapolated to larger geographic areas to imply the consequences of land management, but few studies are specifically designed for that purpose.

Larger scale monitoring studies that track the change in habitat condition over time may allow for the regional detection of responses that are due to changes in management (Urquhart et al. 1998; Larsen et al. 2001). For example, the Environmental Monitoring and Assessment Program (EMAP) has proposed regional monitoring efforts

that attempt to detect change in environmental conditions over the United States (Whittier and Paulsen 1992; Larsen et al. 1995). Change detection requires periodic revisits to the same sites to observe both the direction and magnitude of change. In some cases, the period of time to detect change may take from 5 to 20 years, depending on the design and variables of interest (Urquhart et al. 1998).

Managers often have a need for more rapid assessment of environmental conditions in lieu of repeated visits to the same site. One way to look at differences is to evaluate conditions between sites impacted by some form of management and sites that have relatively intact physical and biological processes. Regional reference sites may also be a way to evaluate the potential of streams within the same region (Karr and Dudley 1981; Hughes et al. 1986). Classification systems have been developed to evaluate the impairment of water quality based on the conditions of reference sites in Australia and Great Britain (Moss et al. 1987; Wright et al. 1993; Wright 1995) and are currently being used to assess water quality condition in the United States (Hawkins et al. 2000). One difficulty with using this approach is that the number and quality of reference sites may be limited in certain locations.

In this study, we examined stream reaches in watersheds representing both managed and reference conditions to determine if we could detect differences in physical habitat variables. This work is part of a broader study to evaluate whether aquatic and riparian conditions are being maintained, degraded, or restored on federal lands administered by the U.S. Forest Service and the Bureau of Land Management in the Columbia basin (Kershner et al. 2004). Our objectives were to describe differences in commonly measured stream habitat variables between reference and managed stream reaches and to evaluate the usefulness of these variables in measuring change in this analysis.

Methods

The study area is located in the interior Columbia basin and includes federal lands within the states of Idaho, Montana, Oregon, and Washington (Figure 1). Data for this study were collected during the summers of 1999–2001 on lands managed by the USDA Forest Service and USDI Bureau of Land Management.

Study design.—We sampled stream reaches in randomly selected watersheds in Idaho, Montana,

Oregon, and Washington, using the selection process outlined in Kershner et al. (2004). Sample watersheds contained greater than 50% federal land and streams were generally fish-bearing and easily wadable at summer base flows. The sample watersheds were sixth hydrologic unit code (Seaber et al. 1987) that represented 62 reference watersheds and 199 managed watersheds. Watersheds were considered reference if there had been no livestock grazing within the past 30 years, less than 10% of the watershed had undergone timber harvest, there was no evidence of mining in proximity to riparian areas, and road density was less than 0.5 km/km². Managed watersheds included a full complement of management activities, including timber harvest, road building and maintenance, livestock grazing, mining, and recreation. Livestock grazing was present on 165 of the managed watersheds and we documented current timber harvest on 49 watersheds. Not all management activities were present in every watershed.

One stream reach was sampled in each watershed. Selected reaches were designated as “integrator” reaches having less than 3% gradient and were always more than 20 times the bank-full width in length, but never less than 80 m (Kershner et al. 2004). Sample reaches were the first reach on federal lands upstream of the watershed outlet to meet these criteria. Potential integrator reaches that were influenced by beaver activity were excluded from sampling. In general, these reach types represented pool-riffle or plane-bed channels that should have the greatest sensitivity to increases in sediment supply and peak flows (Montgomery and MacDonald 2002).

We evaluated differences among a set of commonly used instream, bank, and substrate variables that have been used in other studies of land use effects to characterize reach condition. Field methods for each variable have been reported in Kershner et al. (2004). All field measurements were conducted during the summer sampling season and streams were sampled at base flows.

A set of watershed descriptors was used to characterize conditions for each watershed (Table 1). We used these variables to describe the physical conditions of the watershed and to characterize key aspects of the management history. These descriptors were also evaluated as potential covariates in the analysis.

Analysis.—The sample area represents a large and diverse set of biological and physical characteristics that might influence our ability to detect differences in stream habitat characteristics be-

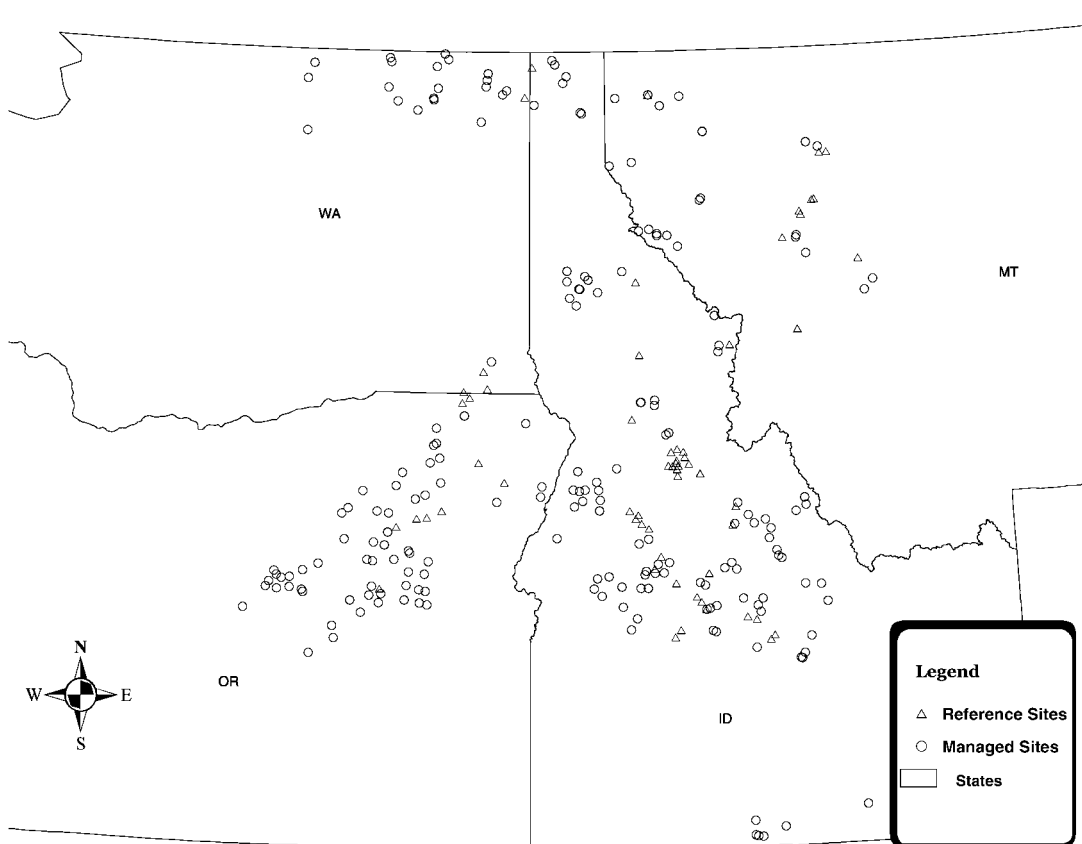


FIGURE 1.—Map showing study area locations.

tween managed and reference stream reaches. Because these differences could potentially confound the relationships we wished to test, we wanted to control the amount of effect those variables could have on the analysis. We used a multistep process

to first identify the most influential control variables and then tested whether the stream habitat characteristics of managed and reference streams were different after adjusting for differences by using analysis of covariance (ANCOVA).

TABLE 1.—Characteristics of reference and managed watersheds within the upper Columbia River basin study area. These variables were evaluated as potential covariates in the analysis. Variables denoted by * are significantly different. Variables denoted by ** were used in the initial determination of reference and managed designations.

Variable	Managed (<i>n</i> = 199)		Reference (<i>n</i> = 62)	
	Mean	SD	Mean	SD
Elevation (m)*	1,655	429	1,756	439
Gradient (%)	1.27	0.7	1.21	0.7
Sinuosity	1.34	0.31	1.42	0.39
Average bank-full width (m)*	5.7	2.8	7.1	4.2
Mean annual precipitation (mm)*	810	302.6	1,017	287.3
Area (km ²)*	39.9	28.6	31.7	26.1
Entrenchment ratio	2.33	0.59	2.41	0.6
Stream density (km/km ²)	1.54	0.65	1.58	0.94
Number of road crossings**	30.0	36.4	0.92	2.2
Road density (km/km ²)**	1.60	1.1	0.08	0.14
Riparian road density (km/km ²)**	0.62	0.55	0.02	0.04
Federal ownership (%)*	95.4	7.7	99.8	0.7

Initially, we believed that some measure of stream size and gradient would be useful predictors for many of our dependent variables, based on previous studies (Lanka et al. 1987; Kozel et al. 1989). We evaluated stream gradient, sinuosity, bank-full width, and entrenchment ratio from our reach-level data set and watershed descriptors and other watershed-level variables as potential covariates (Table 1).

We used a model building procedure described by Milliken and Johnson (2001) to help determine which of the potential covariates explained the greatest amount of variation in the stream habitat variables and hence would be most useful as covariates. The first step of this process was to fit a model of the treatment variable (management type) to the stream habitat variable and compute the residuals. We used the same process for the potential covariates, generating a model of the treatment variable to the covariates and computing the resulting residuals. We used the residual of each potential covariate as an independent variable and the residual of each stream habitat variable as a dependent variable to build multiple regression models.

We used multiple regression models to evaluate the predictive capability of each independent variable for each of the dependent variables and examined all models to determine which of the watershed variables or groups of variables explained the most variation for the full suite of dependent variables (Milliken and Johnson 2002). We used adjusted r^2 and Akaike information criterion (AIC) scores to rank the models to determine which multiple regression model explained the most variability with the fewest variables (Burnham and Anderson 2002). In addition, descriptor variables had to have a well-established mechanistic relationship with the response variables to avoid spurious correlations (Anderson et al. 2001). The final set of covariates used in the ANCOVA was also based on consistency across response variables.

An unbalanced, incomplete-block design ANCOVA was performed on each of the dependent variables by using geology type as the block effect and a subset of the best variables (bank-full cross-sectional width, stream gradient, and average precipitation) as the covariates (Littell et al. 1996). We stratified geologic type into four general categories throughout the basin: igneous, sedimentary, metamorphic, and volcanic.

We performed the ANCOVA by first evaluating whether the covariate slopes were parallel. If covariate slopes were parallel, then evaluation of the

treatment effect (managed versus reference) can occur at one covariate value (conventionally this is the y-intercept). If the slopes are not the same between treatments, evaluation of the treatment effect is still possible, but comparisons should be made across several covariate values because the difference between treatments changes across the range of values for the covariate (i.e., lines are not parallel). To test if the slopes of the regressions between the covariate and the response variable were significantly different between treatments, we set $P = 0.1$.

We evaluated the differences between treatment means after accounting for the block and covariate effects and estimated the 90% CI surrounding the difference (LSMEANS procedure, SAS 1999). We used confidence intervals to evaluate the effect size and the variability surrounding this effect. If a difference of 0 between reference and managed streams is contained within these confidence intervals, then a conclusion of no statistical difference can be made.

In cases where slopes were significantly different between treatments, the differences were evaluated across the range of values for each covariate. For example, significant differences in a variable might be detected where the value of a covariate is small, but become insignificant as the value becomes larger.

Results

Watershed Characteristics

There were distinct differences in the physical characteristics that described reference and managed watersheds. Watersheds containing reference stream reaches had a slightly greater percentage of federal lands (99.8%) and a smaller watershed area (31.7 km²) than watersheds containing managed stream reaches (95.4% and 39.9 km², respectively; Table 1). Watersheds containing reference stream reaches tended to occur at higher mean elevations (1,756 versus 1,655 m) and received more annual precipitation (1,017 versus 810 mm; Table 1). Sample streams within reference watersheds were typically wider (7.1 versus 5.7 m), but showed little difference in stream density, gradient, sinuosity, or entrenchment ratio (Table 1).

There were large differences in the management variables used to segregate reference and managed watersheds. Riparian road density (0.62 versus 0.02 km/km²), road density (1.60 versus 0.08 km/km²), and mean number of road crossings (30.0

TABLE 2.—ANCOVA-adjusted means and standard errors of study variables reported for reference and managed watersheds. Differences are significant ($P < 0.10$) when the 90% confidence intervals do include 0; variables with significant differences are denoted with asterisks. The designation d_{50} is the median particle size, the designation d_{16} the diameter for which 16% of the particles are smaller.

Variable	Reference		Managed		Difference	90% CI
	Mean	SE	Mean	SE		
Residual depth (m)*	0.37	0.015	0.33	0.008	0.04	0.01–0.07
Percent pools	49.8	2.98	48.3	2.38	1.5	–2.4–5.5
Bank stability (%)*	80.3	2.27	74.4	1.24	5.9	1.5–10.2
Bank angle (°)*	100.4	5.53	108.6	5.03	–8.1	–13.3–3.1
Undercut percent*	34.9	4.08	28.4	3.65	6.6	2.6–10.6
Undercut depth (m)*	0.11	0.014	0.09	0.012	0.03	0.01–0.04
Percent pool fines	25.5	3.34	28.6	2.06	–3.0	–9.0–2.9
Percent riffle fines	26.1	3.27	24.4	2.45	1.7	–3.1–6.5
d_{16} (mm)	9.3	1.68	11.5	1.32	–2.3	–4.6–0.1
d_{50} (mm)*	29.7	3.21	38.7	2.24	–9.0	–14.1–3.8

versus 0.96; Table 1) were all substantially higher in managed watersheds.

Covariate Selection

Three descriptive variables consistently explained the greatest amount of variability in the dependent variables used in our analysis: average precipitation, mean bank-full width, and gradient. Other variables occasionally appeared to be as important in some models, but these instances were generally isolated and not important in most cases. We chose the final set of three variables based on the number of occurrences in all models and their relevance as reported in the published literature for other studies. Our final selection represents the most-parsimonious set of variables. All three variables were used in each model unless otherwise indicated.

ANCOVA

Bank descriptors.—Stream banks were more stable in reference stream reaches than in managed stream reaches (80.3% and 74.4%, respectively; Table 2). We evaluated bank stability with an ANCOVA using average width as the single covariate. Gradient and average precipitation were not included in the ANCOVA because slopes were not significantly different from 0. Differences in bank stability between reference and managed streams were statistically significant.

Bank angle and bank undercut were both significantly different between reference and managed reaches. Reference stream reaches had a lower mean bank angle than did managed stream reaches (100.4° versus 108.6°, respectively; Table 2). We observed differences in both the percent and the depth of undercut banks between reference and managed stream reaches. The mean percent of

undercut banks was significantly greater in reference stream reaches (34.9%) than in managed stream reaches (28.4%). The mean undercut depth in reference streams was greater than in managed streams (0.11 versus 0.09 m, respectively; Table 2) and the differences were statistically significant.

Instream habitat characteristics.—Our analysis of the percentage of pools and the residual depth of pools in managed and reference stream reaches yielded mixed results. We observed small differences in the percent pools between reference and managed stream reaches (49.8% and 48.3%, respectively; Table 2), but these results were not statistically significant. Mean residual depth was significantly greater in reference stream reaches (0.37 versus 0.33 m).

The width to depth ratio in reference streams was slightly lower than for managed streams. Because our tests of the covariates revealed that our assumption of equal slopes was violated for bank-full width, we analyzed the differences in width to depth ratios over the full range of bank-full widths. We observed significant differences in the width to depth ratios only when streams were more than 5 m wide (Figure 2).

Surface substrate.—We found little difference in both measures of fine sediment between reference and managed streams. The mean percent pool tail fine sediment was less in reference stream reaches (25.5%) than in managed stream reaches (28.6%; Table 2), whereas the percentage of riffle fine sediments was slightly greater in reference streams (26.1 versus 24.4%; Table 2). Results were statistically insignificant in both analyses.

Two measures of particle size distribution (the diameter for which 16% of the particles are smaller and the median particle size; d_{16} , d_{50}) showed dif-

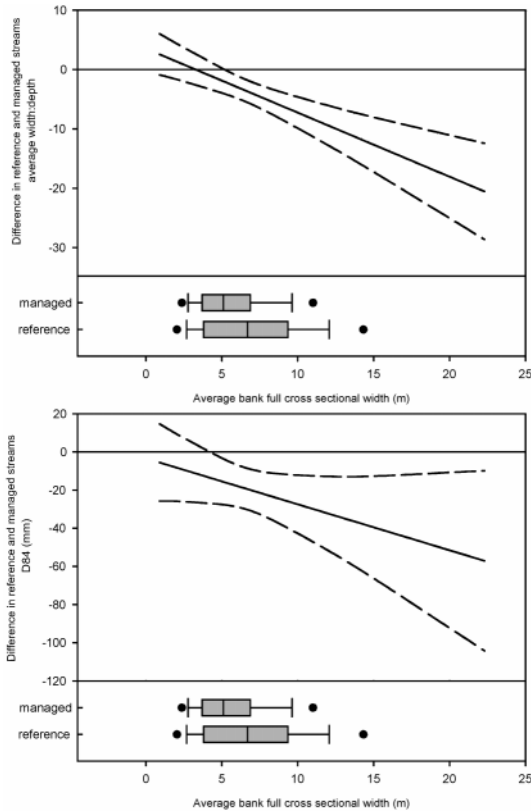


FIGURE 2.—Differences in the width-depth ratio and the d_{84} value (the diameter for which 84% of the particles are smaller) between reference and managed watersheds. The solid lines indicate mean differences for a given bank-full width and the dashed lines 90% confidence intervals; differences are significant where the confidence interval does not include zero.

ferences between reference and managed stream reaches; however, these differences were statistically significant in the ANCOVA only for d_{50} . The d_{16} in reference stream reaches (9.3 mm) was smaller than that in managed stream reaches (11.5 mm), but this difference was not statistically significant (Table 2). The difference in the d_{50} between reference and managed stream reaches was 29.7 and 38.7 mm, respectively (Table 2).

Because our tests of the covariates revealed that our assumption of equal slopes was violated for cross-section width, we analyzed the differences in the diameter for which 84% of the particles are smaller (d_{84}) over the full range of cross-sectional widths. We observed significant differences between reference and managed stream reaches when streams were more than 5 m wide. These differ-

ences became larger as streams got wider (Figure 2).

Discussion

We observed differences in measures of stream habitat between reference and managed watersheds, generally in the direction we expected. For this study, we divided stream habitat measures into three components that have commonly been reported in the literature: changes in stream banks and cross-sectional profile, changes in stream habitat as indicated by differences in the type and amount of channel units and changes in pool depth, and changes in measures of substrate. In general, studies evaluating land use effects on stream habitat have reported differences in all or a subset of these conditions.

Changes in the stream banks and width-depth relationships have been most frequently reported in literature reporting the effects of livestock grazing on stream habitat (Platts 1991; Belsky et al. 1999; Rinne 1999). The results of livestock grazing have been reported as increasing width to depth ratios as a consequence of the removal of riparian vegetation (Platts 1981; Kauffman et al. 1983; Matthews 1996) and the direct, mechanical breakdown of stream banks from livestock hoof damage (Kauffmann et al. 1983; Marlow and Pogacnik 1985). Additional consequences have included the loss of bank stability, reduction of undercut banks and the depth of the undercut, and increased bank angle (Platts 1981; Hubert et al. 1985; Myers and Swanson 1995; Knapp and Matthews 1996).

Although several review articles report results from individual studies (Platts 1991; Belsky et al. 1999; Rinne 1999), few studies have examined changes in stream bank characteristics over a large geographic area. Results from our study suggest that where livestock grazing had been removed for at least 30 years or minimal influence from land management had occurred, we saw significant differences in our bank-related response variables across a broad area. Stream banks were more stable, were more undercut, had deeper undercut banks, and had steeper bank angles in our reference sites.

These results are consistent with other studies where livestock have been removed or excluded. For example, bank stability rapidly improved in Mahogany Creek, Nevada, when cows had been removed for 15 years but took longer to change where improved livestock grazing practices were followed by floods on Summer Camp Creek, a similar stream type (Myers and Swanson 1996). Sim-

ilar differences in bank angle have been observed where cattle have been excluded from stream and riparian areas (Kauffman et al. 1983; Platts and Nelson 1985; Rinne 1988; Knapp and Matthews 1996). Bank angles tend to decrease and the amount of undercut banks increase when livestock have been excluded from stream riparian areas (Platts 1981; Myers and Swanson 1995; Knapp and Matthews 1996). The differences we observed across a broad sample area appear to validate these relationships observed at smaller scales in previous studies.

We observed small differences in the percentage of pools between reference and managed stream reaches. This is in contrast to other studies that have reported much larger changes in the percentage of pools in managed streams affected by severe floods (Lisle 1982), timber harvest practices (Ralph et al. 1994; Woodsmith and Buffington 1996), grazing (Myers and Swanson 1996; Magilligan and McDowell 1997), or a combination of land use activities (McIntosh et al. 2000). There are several potential reasons for this difference. Pool percentage may be a relatively insensitive measure of land use changes in our study streams, but McIntosh et al. (2000) reported significant differences in the percentage of large pools in their analyses of natural and managed streams in the Blue Mountains and Northern Rockies, which correspond to much of our study area.

The dominant pool-forming elements may also be different for most of our study streams. The location of our sample reaches often corresponded to low-gradient willow/sedge riparian areas with minimal large wood instream and little riparian forest. Pools were often associated with lateral scour along banks or against root masses of riparian plants, but there was little large woody debris. Large woody debris was the major factor influencing the formation of pools in streams in southeast Alaska (Woodsmith and Buffington 1996) and the western Cascades (Abbe and Montgomery 1996).

Finally, the ability of our crews to consistently identify and measure pools may be limited. In a previous study, the percent of variation attributable to crew measurement was higher for percentage of pools than for any other measurement (56%; Roper et al. 2002; Archer et al. 2004). This may limit the crew's ability to consistently identify the feature and correctly describe the dimensions of pools. However, given that we observed significant differences in the residual depth of pools between managed and reference watersheds, we expected

larger differences in this measure than we observed.

We did observe significant differences in the residual depths of pools between managed and reference streams. McIntosh et al. (2000) surveyed large pools in streams within the Columbia River basin and found that the depth of "deep pools" decreased compared with findings of surveys conducted in the 1930s. Few other published studies have looked at differences between streams with different management histories (but see Woodsmith and Buffington 1996). In our streams, we observed statistically significant differences in the residual pool depths between reference and managed streams. Residual depths provide a more accurate measure of differences because the depth at the riffle crest provides a benchmark for all residual depth measurements (Lisle 1987; Lisle and Hilton 1992). Residual depth may be an important monitoring tool because differences in bed elevation in the pool can be measured somewhat independent of flow, reflecting true differences in pool depth.

Changes in particle size distribution, particularly changes in measures of fine sediment, have often been used to characterize land use effects on fish habitat in streams (Everest et al. 1987; Hicks et al. 1991; Potyondy and Hardy 1994). Sediment-related land use effects may include an increase in the percent of fine sediments in spawning areas (Platts and Megahan 1975; Everest et al. 1987), which could potentially influence the spawning success of salmonids (Tappel and Bjornn 1983; Chapman 1988). Pool-tail fine sediment has been considered a useful indicator to assess the condition of potential spawning areas for many salmonid species within our study area (Overton et al. 1995).

The direction of the differences we observed in pool-tail fine sediment between reference and managed streams is generally consistent with other studies, but the results were not statistically significant. One possible explanation for this is that the variability associated with observer measurement and the measurement sites may have been high and affected our ability to detect differences from management. In a related study, Archer et al. (2004) estimated that 1,000 samples would be needed to detect a 10% difference between managed and reference sites. Given our sample sizes, this could limit our ability to detect differences. We observed less fine sediment in pool tails in reference streams. Fine sediment declined in two Nevada streams after either cattle were excluded

or livestock management changed over a 15-year period (Myers and Swanson 1996). Conversely, Knapp and Matthews (1996) were unable to detect meaningful differences in the percentage of fines in cattle-excluded areas and in grazed areas. They speculated that the amount of time since the enclosures were constructed may not have been long enough to see measurable changes.

Relatively few studies have examined the differences in other measures of particle size distribution between reference and managed streams. MacDonald et al. (1991) recommend incorporating multiple measures of the particle size distribution (d_{16} , d_{84}) as well as the median particle size to more accurately portray the variation in particle size distribution. In this study, we saw differences in median particle sizes in all sizes of streams and differences in the d_{84} where streams were more than 5 m wide. We expected the median particle size to be larger in reference streams where the expected percentage of fine sediments was less, but this was not the case. Potyondy (1990) observed differences in the median particle size both before and after a large wildfire in Boundary Creek watershed. Median particle size before the fire corresponded to gravel prefire, shifted to sand by 1 year postfire, and returned to gravel within 2 years after the fire. Little difference was apparent in streams that experienced low- to moderate-intensity burns.

The location of sample sites within a watershed is important if we are to detect changes in stream habitat due to management. For example, Montgomery and MacDonald (2002) recommend that stream monitoring programs must consider differences in sensitivity and response related to channel type, spatial and temporal variability in inputs, and the effects of other controls at the reach and watershed scale. We chose our sample locations to reflect areas that integrate the conditions of the watershed and could potentially be responsive to changes in management and selected variables that had been shown to be sensitive to change. These locations corresponded to two different channel types—pool-riffle channels and plane-bed channels (Montgomery and Buffington 1997), both of which are considered relatively sensitive to inputs of sediment and peak flows. Although we detected changes in some variables, we did not get an expected result in others. This may be a function of sample location or the possibility that these variables are relatively insensitive to management effects.

Analysis of covariance is an effective way to

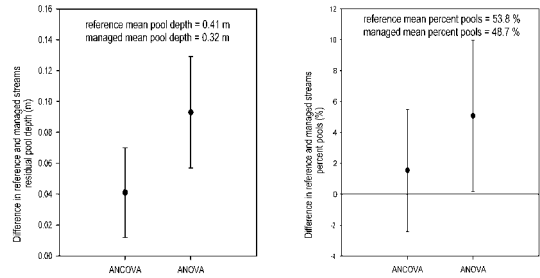


FIGURE 3.—Differences in two variables pertaining to pool characteristics (residual pool depth and percent pools) in reference and managed watersheds according to ANCOVA and ANOVA. The points indicate mean differences and the vertical lines 90% confidence intervals; differences are significant where the confidence interval does not include zero.

compare data across a large, relatively heterogeneous landscape where sample site stratification may be impractical or sample sizes are limited. In our analysis, we were able to identify three variables (average precipitation, mean bank-full width, and gradient) that were useful in partitioning and potentially reducing the variability associated with a large sample area. By identifying the source of this variability, we were able to more precisely determine the true differences in most variables. For example, our analyses of the differences in percentage of pools revealed no statistically significant differences when analyzed with covariates, whereas a simple analysis of variance (ANOVA) indicated a much larger, statistically significant difference (Figure 3). Conversely, both approaches found significant differences when we analyzed differences in residual pool depth. However, comparisons of AIC scores between ANCOVA and ANOVA indicated that for all but one response variable we were better able to explain the variability by including the additional variables used in the ANCOVA. Results that are reported without accounting for important sources of variation may be misleading, especially in sample areas with considerable heterogeneity.

The use of confidence intervals to evaluate the size of an effect and the variability around the effect provided a useful way to explore the extent of differences between treatments at different values of the covariates. For example, a simple analysis of the difference in width to depth ratios as they covaried by bank-full width would reveal no significant difference. However, the use of covariates and confidence intervals allows for the exploration of these differences at a range of bank-full widths and provides useful insights into the

size of those differences as bankfull widths increased.

The development of reference conditions or reference sites has been suggested as one way to evaluate the influence of human impacts on streams and associated biota. Definitions of reference conditions vary but often imply "minimal disturbance" (Hughes et al. 1986; Bauer and Ralph 2001). Reference conditions are often used to evaluate the degree of impairment to biotic communities in the United States (Karr 1981, 1991; Karr and Chu 1997) and abroad (Wright et al. 1993; Norris 1996; Moss et al. 1999). Implicit in many of these studies is the idea that streams have some desired structure and function based on acceptable norms. However, recent literature suggests that the distribution of conditions expressed across the landscape, even in landscapes with minimal human intervention, is highly variable (Reeves et al. 1995).

The variety of conditions that exist across the landscape of the Columbia River basin represents a history of natural disturbances such as wildfire, flooding, insect infestation, and climate change (Hessburg and Agee 2003). Stream channel conditions are artifacts of the frequency and magnitude of disturbance at a variety of scales. For example, forest succession and wildfire frequency are linked to the frequency of erosional events and the postfire sedimentation that structure stream channels (Meyer and Pierce 2003). These events result in a mosaic of stream habitat conditions across the broad landscape and at local scales (Benda et al. 2003). Consequently, the expression of conditions in reference sites should represent a range of values, from what we might consider the "worst" to the "best."

Our definition of reference conditions in this study makes no attempt to set artificially high values for any parameter. Reference conditions simply reflect the current state of these systems without adjusting for conditions that we perceive may not be optimum. In this case, the distribution of conditions represents the evaluation criteria in our analysis and we should be able to measure the adjustment in our study streams if management influences are minimized (Noon et al. 1997).

In conclusion, there is an increasing need to describe the state of aquatic systems across broad landscapes to report on the progress of agency programs to protect aquatic resources. Studies that project the improvement of these resources over time will provide important insights into the success of these programs. In the meantime, managers

require more timely reports as to the status of resources in the absence of trend data. We believe that comparing reference conditions to conditions across managed landscapes provides a credible way to report on the condition of these systems and that the analyses outlined in this paper may be one of many ways to report on that progress.

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